Severe coral bleaching in 2002–2003 at Cobourg Marine Park, Northern Territory, Australia

VICTOR E. GOMELYUK

Marine Biodiversity Group, Biodiversity Conservation,
Department of Natural Resources, Environment and Arts, ATRF,
PO Box 41775, Casuarina NT 0811 AUSTRALIA
victor.gomelyuk@nt.gov.au

ABSTRACT

This study analyses data from six years of monitoring live coral cover of shallow fringing reefs at Garig Gunak Barlu National Park (Cobourg Peninsula, Northern Territory, Australia). The chosen monitoring technique (digital live percent area cover assessment within permanent quadrats) provided high accuracy and had sufficient power to detect both negative and positive changes in coral cover. However, the magnitude of changes after severe coral bleaching from November 2002 to January 2003 showed that the monitoring design, while being appropriate for tracking small disturbances in coral cover, may be superfluous in situations where catastrophic changes follow major coral bleaching events. Four months after the 2002/2003 bleaching, losses in overall live coral cover at different monitored sites (ranging from 42% to 90%, including between 75% and 96% losses of *Acropora*) resulted in serious alterations in the composition of local coral reefs and a decline in reef complexity.

KEYWORDS: Coral monitoring, percent live cover, coral bleaching, digital image analysis.

INTRODUCTION

Garig Gunak Barlu is the first and so far the only area in the Northern Territory where coral reefs are formally protected. The site is one of 24 in the world (and one of five sites in Australian waters) that is remotely observed for coral bleaching by the National Ocean and Atmospheric Administration's (USA) Coral Reef Watch (CRW) Satellite Bleaching Alert (SBA) system. This six-year study is the first comprehensive study of coral reefs in Garig Gunak Barlu National Park.

Coral bleaching is an event related to loss of symbiotic zooxanthellae, a single-cell alga (*Symbiodinium*) in coral polyp tissues, and subsequent coral death (Warner *et al.* 1999). Coral reef bleaching is a general response to stress that has severely affected reefs in tropical waters throughout the world. It can be induced by a variety of factors, alone or in combination. Higher-than-normal sea temperature (Glynn 1984, 1988, 1993; Atwood *et al.* 1988; Buddemier and Fuatin 1993; Berkelmans and Oliver 1999) and exposure of reef flat corals to atmosphere and solar irradiance during extreme low tides (Jaap 1979, 1985; Brown *et al.* 1994) are major factors causing coral bleaching. It is estimated that 20% of the world's coral reefs have been effectively destroyed and show no immediate prospects of recovery (Wilkinson 2004).

In 1998 a global coral bleaching event seriously damaged 16% of the world's reefs. Approximately 60% have not recovered significantly or have not recovered at all. During local bleaching events in 2000 and 2003, 20% of coral reefs in Australia and Papua New Guinea were destroyed, or brought to a critical or threatened stage (Wilkinson 2004).

The coral reefs at Cobourg Peninsula are shallow and quite peculiar. Water turbidity, because of a high amount of suspended particles, is relatively high throughout the year. Visibility on coral reefs from May to October rarely exceeds 1–1.5 m; from November to April it ranges between 2.5 m and 4 m. Due to the turbidity, the photic zone for hermatypic corals is relatively thin, and well-developed fringe coral reefs within the Park boundaries are found only in shallow waters up to 1.5–2.5m (spring tide). These coral reefs therefore cannot extend to deeper waters because of a lack of light for the photosynthetic zooxanthellae. As a result, corals are exposed to elevated water temperatures during extreme low tides and 'hot spot' intrusion (Goreau and Hayes 1994). This makes the local reefs of Cobourg particularly vulnerable to coral bleaching. If mass mortality takes place from bleaching, the absence of deeper dwelling corals at Cobourg may also hinder the re-settling and sexual reproduction that aids reef recovery. It is thought that the presence of deeper water corals may act as a 'reproductive refuge' – a source of coral larvae for the reef when impacted by severe coral bleaching.

Due to the crucial role that coral reefs play in the inshore marine environment, coral monitoring is an important tool for examining coral condition, and is also an irreplaceable

tool to assess the condition of marine biota *per se*. Live coral cover estimation as a coral health parameter was first proposed by Loya (1972). A significant reduction of live coral cover is an unambiguous sign of general disturbances in coral community (Loya 1972). Changes in coral cover can be monitored by direct measurements of changes in live coral cover (Endean and Stablum 1973; Done 1981, 1992; Gittings *et al.* 1990; Porter and Meier 1992).

Permanent sites are recommended for long-term monitoring because they offer the greatest amount of information, consistency, repeatability, and reliability (Ohlhorst *et al.* 1988; Rogers *et al.* 1994; English *et al.* 1997). They give a higher statistical power than temporal sites (Brown *et al.* 2004) and are useful for observing specific coral colonies over time. They also provide a precise measure of percentage cover. Photo-quadrats, in particular, provide a permanent record of the benthic communities; the images can be digitised and a very accurate percentage cover estimate can be obtained. Data then can be used to compare fine-scale changes in benthic communities through time. Like all methods, permanent quadrats have their drawbacks: they cannot be used to measure reef rugosity (spatial relief) (Ohlhorst *et al.* 1988) and plate-shaped corals tend to be over-represented relative to columnar-shaped corals. Additionally, this method is difficult to use in areas dominated by fragile branching corals (English *et al.* 1997). However, this method greatly reduces field expenses and time spent under water compared to visual methods, particularly with the use of digital equipment. Computer software can be used to calculate a very precise percentage cover and is the most accurate method available to date (Hill and Wilkinson 2004).

The aims of this study are to implement a monitoring design suitable for the local environment, assess changes to coral cover at reefs within the Park and compare remote coral bleaching prediction from Coral Reef Watch (Strong *et al.* 2004) with situations observed *in situ*.

MATERIALS AND METHODS

Sites description. Study sites are located at Garig Gunak Barlu National Park, Cobourg Peninsula. Sixteen monitoring stations were established at four sites with well-developed coral reefs around the entrance of Port Essington Bay in 2001. These are:

Coral Bay (11° 11' S, 132° 03' E), a very shallow coral reef almost fully exposed during extreme low tides. Coral biodiversity in this bay is very low and the small monitored reef consists of sub-massive *Fungia, Montastrea, Porites, Favia, Favites* and encrusting *Merulina* corals growing on rocks and fragments of dead and partially resorbed plate-like colonies of *Acropora*. Currently, live plate-like and branching *Acropora* are not found in Coral Bay.

Ungalwik (11° 08' S, 132° 08' E), the bay between Black Point and Smith Point and at sites to the south-east

(11° 07' S, 132° 11' E) and south-west (11° 07' S, 132° 11' E) of Sandy Island No. 1 (Fig. 1). Coral biodiversity at all three sites in 2001 was very high and corals were represented by a variety of growth-forms.

The tidal range in the Park waters is up to 2.5 m (spring tide). The highest sea temperatures are observed in November to January (28°C–30.5°C) and the lowest in August (25°C–26°C). Where necessary, water temperature *in situ* was measured in this study using a Horiba U-10 water quality checker.

Anthropogenic factors such as pollution, high nutrient loadings, over-fishing and anchor damage are negligible.

High water turbidity, storms and high temperature events are probably the main factors affecting coral distribution at Cobourg, giving shape to reefs and altering species composition. Water in Coral Bay is particularly turbid at all times of the year because the bay is exposed to strong easterly and north-easterly wind and wave disturbances. Therefore, visibility in Coral Bay rarely exceeds 1.5–2 m.

The very thin photic zone at Cobourg restricts the distribution of fringe coral reefs. Also, corals are found in very shallow waters at a depth of 1.5–2.5 m (spring tide) in locations where wave action is minimal and visibility does not drop below 1–1.5 m throughout the year. Only solitary coral colonies that prefer deeper water, like *Fungia, Montastrea* and massive forms of *Porites, Favia, Favites* and *Platygra* ('bommies'), are located at the lower limit (deeper than 3–4 m) of the photic zone. They comprise less than 1 % of the total coral reef area.

Monitoring design. Each monitoring station consisted of four 1x1 m quadrats making a plot of 4 square metres (fixed photo quadrat method, English *et al.* 1997; Green and Smith 1997; Rogers *et al.* 1994). Since different coral
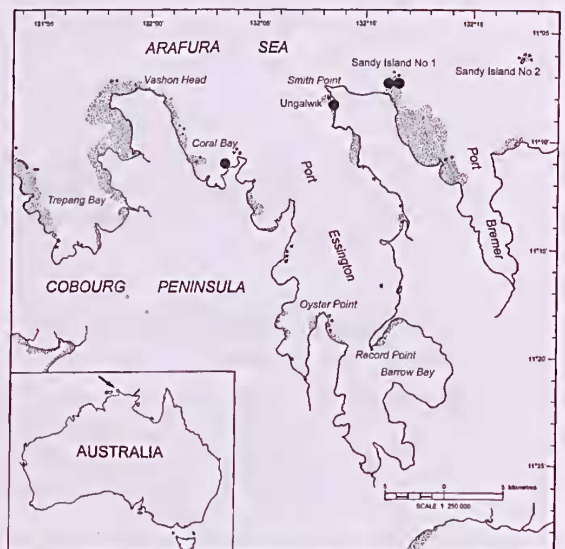

Fig. 1. Regional map and four study sites (indicated by dots) at Cobourg Peninsula, Garig Gunak Barlu National Park.

species have a different tolerance toward adverse external factors, reefs where one coral life form was dominant were avoided. The position for each monitoring station was randomly selected within a previously checked reef. Corners of each permanent quadrat were marked with stainless steel pegs.

Poor underwater visibility created problems with capturing clear images. To obtain satisfactory bottom images it was essential to reduce the distance between the photographed object and the camera lens. To achieve this each 1 m² quadrat was divided into four subplots before taking a photo. An 8 mm white rubber band was stretched around all four pegs to sit on them without arching; then, thinner white rubber bands were stretched between the rectangle's sides creating four subplots about 25 000 cm² each. This, combined with the use of Amphibico® 80 degree 'wet' wide-angle lens, helped to reduce the distance between the camera and the bottom surface to less than 1.5 m. Still digital images were taken using a Sony® TRV 120E video camera in an Amphibico® Dive Buddy housing equipped with a 52 mm internal URPro® filter for optimum colour correction. Making still digital images of each monitoring site (four 4 m² quadrats) required 45–60 minutes.

Digital images were frame-grabbed using Adobe Premier 6.0 software and then enhanced, colour corrected and adjusted using Adobe Photoshop 7.0 and saved according to subplot coding. Photographs were digitised and archived on CD-ROMs.

Data analysis. During image analysis, the area of the whole subplot was manually selected and the area value assessed using ImagePro Express software (Fig. 2A). Then all live coral colonies (and part of colonies inside the subplot) were selected (Fig. 2B, C) and their area assessed. The software automatically calculated subplot areas and areas of all live coral colonies. Data were then exported to MS Excel and percent live coral cover was calculated using the formula:

Subplot percent live coral cover = (Σ live coral colonies cover area*100)*Total subplot area^{-1}

The results are based on live coral percent cover data assessed five times during the period April 2000 to May 2006. No assessment was made in 2004. In March 2005 tropical cyclone Ingrid destroyed all four stations at the south-east site of Sandy Island No 1. Subsequently, only 192 subplots have been used for analysis. Only visual transects were made to assess the reef at this site in 2005 and 2006. In total, 320 subplot images (replicates) from
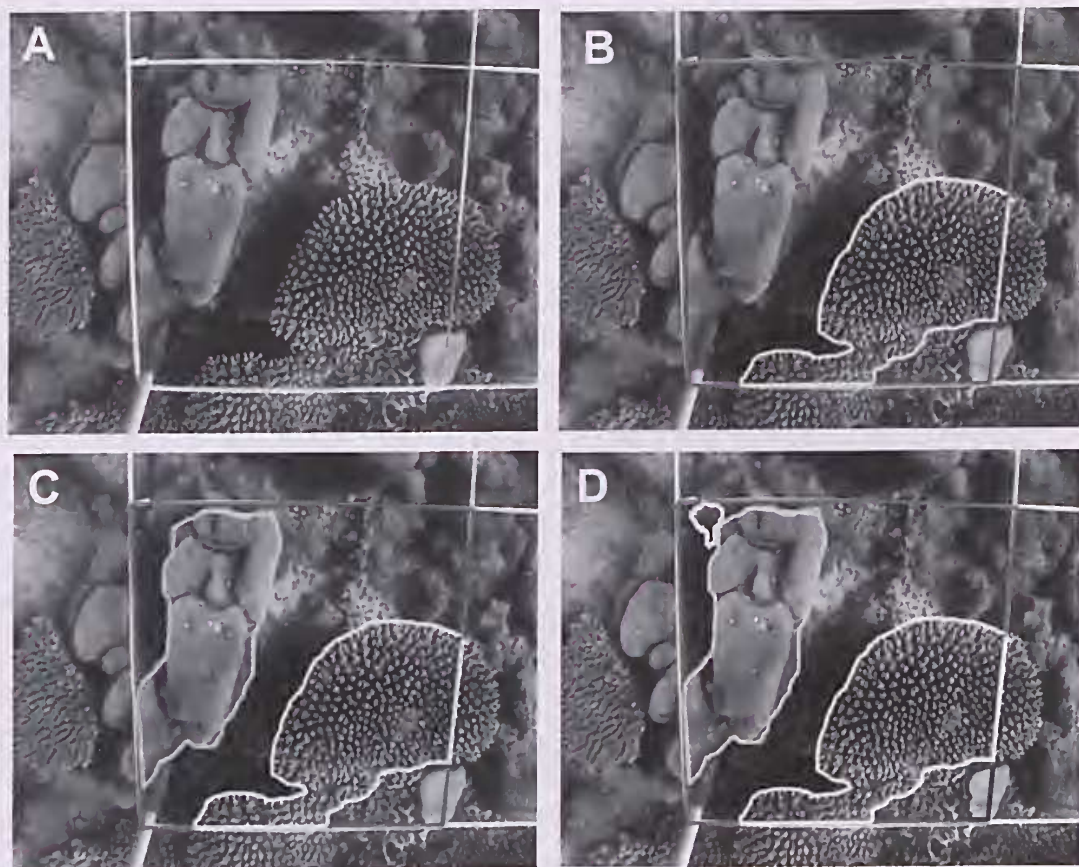
Fig. 2A–D. Different stages of live coral colonies selection process during live percent cover assessment using ImagePro Express 4.0 software.

Table 1. Results of ANOVA comparison of changes in live coral cover within monitored sites at Cobourg from 2001 to 2006. *df,* degrees of freedom; SS, sum of squares; MS, mean square; *F,* F-Ratio; NS, non significant.

[1] Coral cover changes expressed as a fraction with numerator – overall coral cover, denominator – cover of *Acropora*; negative value indicates coral cover decline, positive value indicates increase in coral cover; [2] No *Acropora* in study area of Coral Bay.

	2001–2002	2002–2003	2003–2005	2005–2006
Coral Bay				
ANOVA of coral cover changes		*df*=4; SS=4.87; MS=1.21; *F*=80.39; *p<0.001*		
Coral cover changes[1,2]	-2.3	-80	2.6	5
HSD Tukey comparisons	*p=0.92* NS	*p<0.001*	*p=0.19* NS	*p<0.001*
Ungalwik				
ANOVA of coral cover changes		*df*=4; SS=8.82; MS=2.20; *F*=50.29; *p<0.001*		
Coral cover changes[1]	$\frac{-0.7}{-4.1}$	$\frac{-52.3}{-76}$	$\frac{-5.3}{-3.2}$	$\frac{4.9}{2.1}$
HSD Tukey comparisons	*p=0.90* NS	*p<0.001*	*p=0.44* NS	*p=0.72* NS
South-east side of Sandy Island No 1				
ANOVA of coral cover changes		*df*=2; SS=23.48; MS=11.73 *F*=251.19; *p<0.001*		
Coral cover changes[1]	$\frac{0.3}{1.7}$	$\frac{90.3}{96.5}$	-	-
HSD Tukey comparisons	*p=0.87* NS	*p<0.001*	-	-
South-west side of Sandy Island No 1				
ANOVA of coral cover changes		*df*=4; SS=13.58; MS=3.39; *F*=74.44; *p<0.001*		
Coral cover changes[1]	$\frac{0.7}{1.2}$	$\frac{-42.6}{-88.8}$	$\frac{-11.2}{-100}$	12.1
HSD Tukey comparisons	*p=0.98* NS	*p<0.001*	*p<0.001*	*p=0.003*

each of the remaining three monitoring sites were used for comparison using single factor ANOVA. Live coral percent cover values within each site were dependent variables and years when the observation was made were the factor. Percents data were ARCSINE - transformed prior to the analysis.

Two types of power analysis were undertaken using STATISTICA 6.0 and Monitor version 6.2. These incorporated:

1. An assessment of statistical power vs. group sample size. Changes in live coral cover with the chosen monitoring design could be detected with sufficient statistical power (>0.8), and

2. An assessment of the power to detect trends (percents of changes) in live coral cover. According to the model (Gibbs and Melvin 1993), the statistical power would be sufficient (>0.8 power) to detect ≥ 3% positive and > 4% negative changes in coral cover.

RESULTS

ANOVA comparison of changes in live coral cover within sites from 2001 to 2006 revealed significant decline in overall coral cover at all sites corresponding to a substantial decline in *Acropora* cover (Table 1, Fig. 3A–D). All of these changes were predominantly the result of the coral bleaching event that occurred in the Park between November 2002 and January 2003.

The first recorded coral bleaching event occurred at several sites at Cobourg in November 2001. Some sites (coral reefs at the south-east and south-west of Sandy Island No. 1) were not affected. Corals were exposed to elevated temperatures (mean 32.3° C, maximum 32.6°C) at impacted sites (Coral Bay and Ungalwik) for 7 days. A 'hot spot' was relatively thick. A temperature of 32.3°C was recorded in both Coral Bay and Ungalwik at a depth of 3–3.5 m. Mass coral bleaching occurred at these two sites and up to 80% of all observed coral colonies were affected. Corals species more sensitive to bleaching, such as *Seriatopora hystrix* and *Stylophora pistillata*, died and are now extremely rare at both sites. However, the majority of other corals quickly recovered and regained their normal colouration within 2 to 3 weeks. The impact of this bleaching event in 2001 was thought to be minimal at affected sites. Changes in live coral cover between 2001 and 2002 were statistically not significant at all monitoring stations (Table 1, Fig. 3 A–D).

Far more serious changes were recorded after the catastrophic coral bleaching at all sites from November 2002 to January 2003. This bleaching was caused by a combination of prolonged elevated sea water temperature (31.5°–32.2°C), well beyond the coral bleaching threshold, and local heating caused by poor water circulation in shallow reefs during extreme low tides (water height from 0.1m–0.4 m), occurring during periods of elevated insolation during hours around midday (11:00–14:00). Such conditions developed at the beginning and at the end of November 2002 (five 1.5–2 hour episodes), in December 2002 (nine episodes) and in January 2003 (four episodes). Water temperature at shallow reefs reached 35–36°C during this period. In addition, during extreme low tides, coral reefs were exposed to the direct heat of the sun for periods of 0.5 hour to 1 hour. According to the Australian
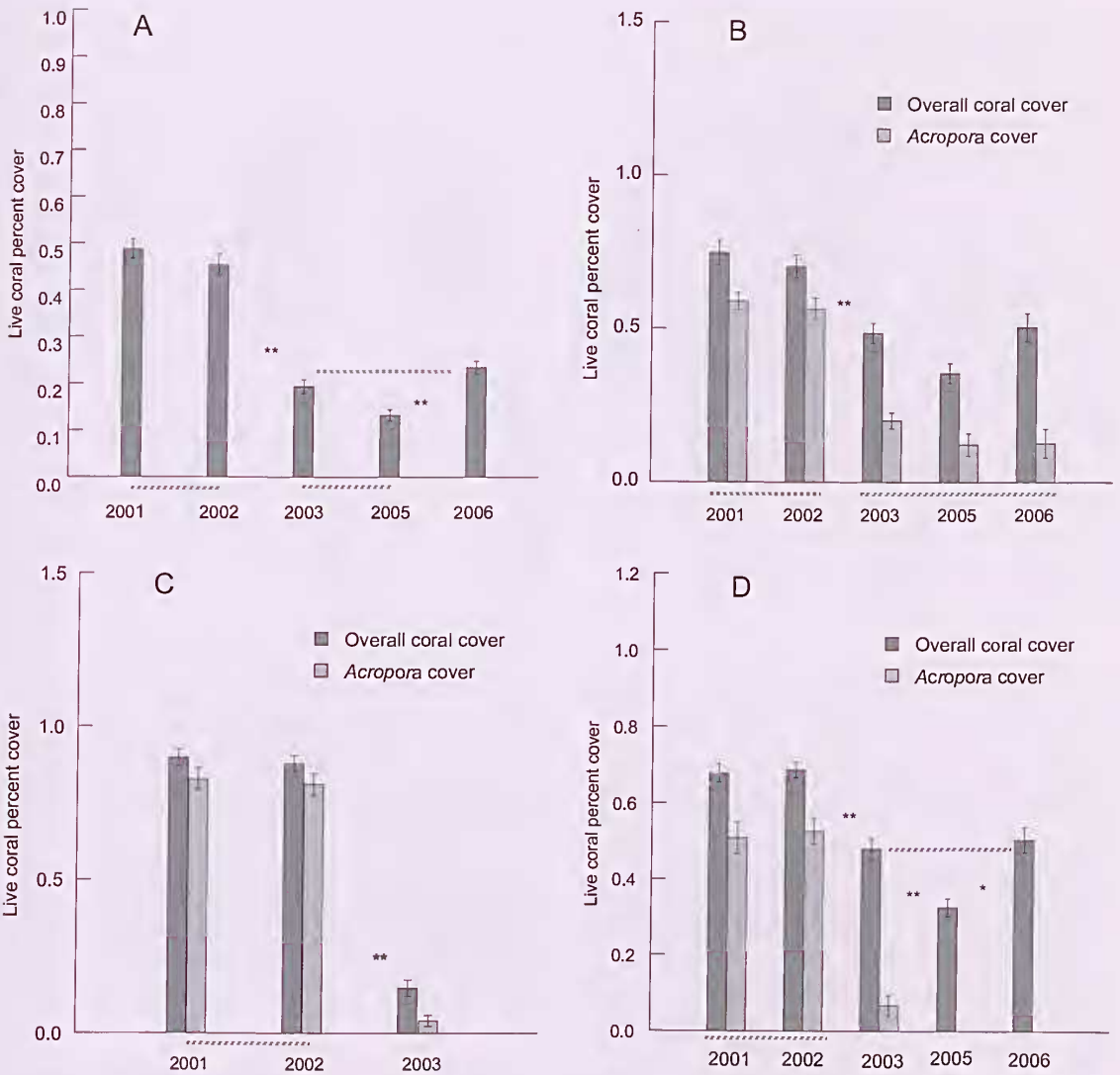
Fig. 3 A–D. Changes in mean live coral cover at monitoring stations during the study. Vertical axes – mean live coral percent cover, ARCSIN transformed. Horizontal axes – years. Vertical bars are ± 1 standard error. Values connected with dashed line do not differ at p>0.05 level (Tukey HSD multiple comparison). Asterisk between columns indicates significant difference at p<0.005 level and two asterisks – at p < 0.001 level. A, Coral Bay; B, Ungalwik site; C, south-east site at Sandy Island No. 1 (monitoring stations at this site were destroyed by tropical cyclone Ingrid in March 2005); D, south-west site at Sandy Island No. 1.

Bureau of Meteorology, the weather in November and December 2002 was very hot and still, with intense solar radiation and absence of clouds. As a result, almost 100% of all coral colonies were partially or completely bleached. Soft corals, sea anemones, giant clams and other invertebrates susceptible to bleaching were also affected. After 18 January 2003, intense storms and rain decreased the water temperatures to 27.5–28°C.

By 13 February 2003 approximately 80% of corals remained bleached. Approximately 30% of coral colonies (mainly *Acropora*) were already dead and covered with green filamentous algae. An assessment of the damage to coral reefs was conducted in April 2003, four months after the bleaching.

The relatively shallower Coral Bay site where *Acropora* was not present lost 80% of its overall coral cover (non-transformed percent data, Table 1). Live cover had significantly declined at all monitoring stations (Table 1, Fig. 3A–D).

An assessment conducted in 2005 indicated there was no statistically significant decline nor an improvement in the live coral cover at Coral Bay and at Ungalwik
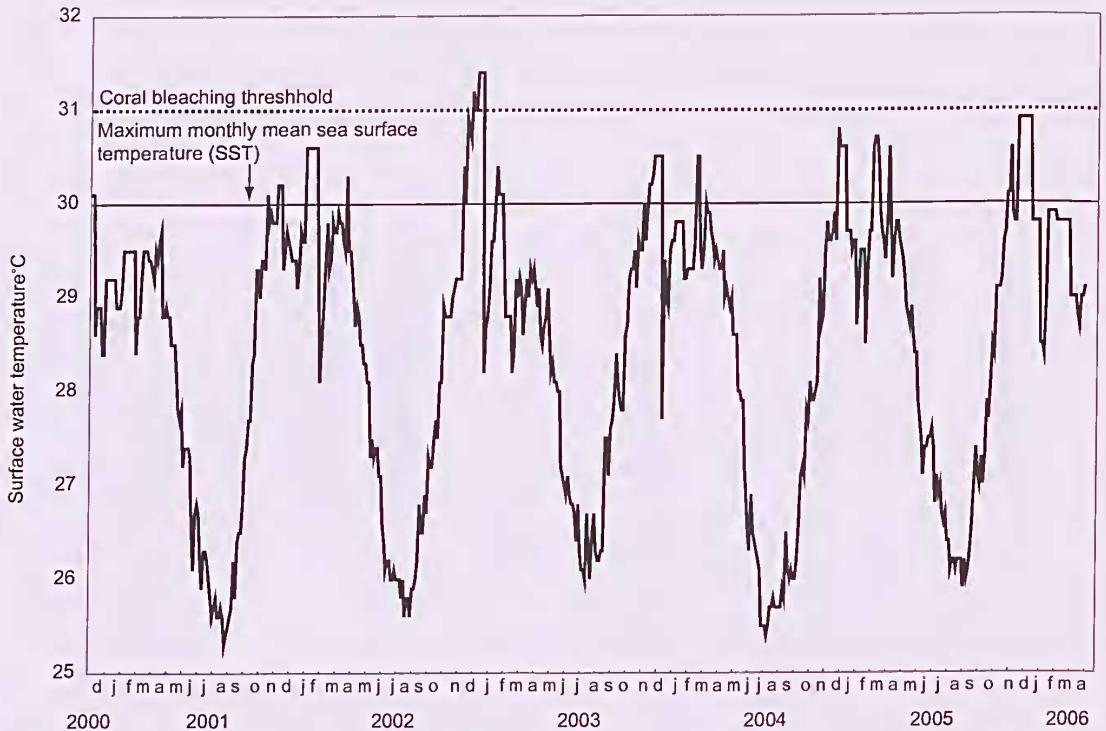
Fig. 4. Satellite data on annual surface water temperature coral thermal stress at Cobourg Peninsula for December 2000–April 2006. Horizontal axes, years and months. Vertical axes, surface water temperature. Data courtesy of the National Oceanic and Atmospheric Administration, Coral Reef Watch. Data is available from: http://coralreefwatch.noaa.gov/satellite/current/sst_series_cobourg_cur.html

(Fig. 3A–D). However, at the south-west site at Sandy Island No. 1, the live percent cover had significantly dropped, and *Acropora* corals had completely disappeared from the monitored stations (Table 1, Fig. 3A–D).

An assessment conducted in 2006 suggested there were definite signs of non-acroporid corals cover recovery at monitoring stations at Coral Bay (Fig. 3A) and at the south-west site at Sandy Island No. 1 (Fig. 3D). At Ungalwik non-acroporid live coral cover values did not change significantly (Fig. 3B).

Change in live coral cover was only one of multiple effects from bleaching on coral communities in 2002–2003. Other effects included serious alterations in the reef composition. Before 2002, table, plate and branching forms of *Acropora* corals dominated at reefs at three monitored sites (but not at Coral Bay reef where these taxa were not found during the baseline study). The decline in live coral cover recorded at these sites after bleaching, predominantly reflects depletion of *Acropora*. The most serious reef composition alterations occurred at sites around Sandy Island No. 1 where colonies of Acropora remain in a very small quantity and only in deeper areas of the reef. At Ungalwik, the bleaching impact was reduced and a decrease in *Acropora* correlated with this comparatively lower decline in overall live coral cover

(Fig. 3A–D). Hence, despite the increased severity of heat and sun exposure at this reef from November to December 2002, the impact of bleaching and live coral cover decline was considerably lower than at other sites (Fig. 3A–D).

DISCUSSION

Goreau et al. (2000) describing 1998 global coral bleaching noted that "in many of reported cases … high turbidity appears to have protected corals, to some degree, from severe mortality." During both bleaching episodes in the waters of Cobourg, high water turbidity apparently gave very little or no protection to corals because bleaching was caused by the 'hot spot' intrusion in the area (2001) and the 'hot spot' intrusion was combined with the corals' exposure to the direct heat of the sun for long periods during extremely low tides (2003).

The timing of the two observed bleaching episodes at Cobourg and the data on satellite-derived highest sea surface temperature (SST) (Fig. 4) suggest that time patterns of coral bleaching at Cobourg are quite distinctive. Maximal SST usually recorded in November and December, during the 'build up' period. 'Hot spot' intrusions at Cobourg occured in November 2001 and in November, December and January 2002–2003. Lowest

astronomical tides typical for this time of the year comprise the other factor increasing the likelihood of coral bleaching.

For other coral reef sites in northern Australia such as Ningaloo Reef (21°30'S, 114°0'E) and Scott Reef (14°30'S, 122°0'E), Indian Ocean; Davies Reef (19°0'S, 147°30'E) and Heron Island (23°30'S; 152°00'E), Coral Sea, Pacific Ocean, both SST and probability of coral bleaching generally increase in January, February and March. The destructive coral bleaching event on the central Great Barrier Reef (GBR) in 1998 started in late January and intensified by late February/early March 1998 (Berkelmans and Oliver 1999). The most intensive and extensive coral bleaching ever recorded at GBR also commenced in January and intensified in the beginning of February 2002, although there was some variation in bleaching intensity and temporal patterns between inshore and offshore reefs (Berkelmans et al. 2004).

There is an obvious dissimilarity in temporal patterns and severity of coral bleaching at Cobourg and at GBR. The worst bleaching on record on the GBR in January and February 2002 (Berkelmans et al. 2004) was preceded by a very short bleaching episode with minimal impact at Cobourg in November 2001. In contrast, there was almost no recorded significant coral bleaching at GBR (and any other reefs in Australian waters or the Indian and Pacific oceans (Wilkinson 2004)) from the end of 2002 to the beginning of 2003, while Cobourg reefs suffered from the severe bleaching that lasted from November 2002 to January 2003.

Coral bleaching in 2002–2003 resulted in serious alterations to coral reefs at the monitored sites at the Park. It caused a substantial drop in relatively high pre-bleaching coral cover. Coral biodiversity and reef topographic complexity decreased substantially because of extensive loss of large table, plate-like and branching forms of colonies of *Acropora*. Coral reefs now predominantly consist of encrusting, sub-massive and foliose forms of coral, including genera, such as *Porites, Favia, Platygra, Goniopora* and *Favites* that are relatively more bleaching-resilient. Apparently, corals at Coral Bay are exposed to more harsh adverse environmental factors (elevated temperature, direct heat of the sun) compared to other sites. The shallow reef at Coral Bay is regularly fully exposed to atmosphere and solar irradiance during low tides and only a small number of more resilient coral species can survive in such conditions. The future of this reef which used to be rich and contained some *Acropora* in the 1980s (H. Larson pers. comm.) is uncertain.

At Ungalwik and at Sandy Island No. 1, table and branching colonies of *Acropora* provided habitat for many coral reef fish and invertebrates, and were the primary reef building corals at Cobourg when this study commenced. This taxon is reported to be a short-lived and fast-growing opportunistic coral group. *Acropora* have high metabolic rates (Jokiel and Coles 1974), and are most susceptible to

physical and chemical disturbance (Van Woesik 1992). They are the least tolerant to bleaching, when compared to such taxa as *Favites, Pocillopora* and *Potrites* (Brown and Suharsono 1990; Gleason 1993) and also have a lower recovery rate (Hoegh-Guldberg and Salvat 1995). According to the records of coral bleaching-caused mortality on the GBR in Australia, *Acropora* mortality was the highest (Wilkinson 2000, 2004).

In 1998 and 2002 destructive coral bleaching at GBR inshore reefs bleached more intensively and extensively compared to offshore reefs (Berkelmans et al. 2004). The coral reef system at Cobourg consists only of inshore shallow fringe reefs. Estimates of the length of time needed for reef recovery range from 10 to 30 years (Hughes 1994; Connell et al. 1997; Done 1999). Possible reduction in the reproductive output of those colonies that survived bleaching, and the lack of colonies in deeper areas, suggest that recovery of the Cobourg reefs to their original state is unlikely. The disappearance of these corals, which play a structural role in the local coral communities and provide an essential habitat for many reef-dwelling organisms, could deplete the biota complexity and biodiversity of the Park.

Elevated natural turbidity and sedimentation are thought to be adverse factors decreasing corals growth (Ginsburg and Glynn 1994; Meesters et al. 1998) and calcification rates (Crabbe and Smith 2002). Severe impact by sedimentation can lead to the suffocation of corals resulting from reduced light penetration (Crabbe and Smith 2002). However, the relationship of turbidity and sedimentation to corals health is far more complex. It was found that corals in Pulau Kubur on the north-western coast of Java, Indonesia, may profit in some way from the turbid waters, for example, by digesting sediment particles (Antony 1999). However, earlier it was proposed that corals at Pulau Kubur have obtained a different strain of zooxanthellae that are more efficient at the ambient irradient levels (Chang et al. 1983; Rowan et al. 1997). Meesters et al. (2002) suggested that more detailed studies are needed to answer the question of why corals in this area are performing better under increased sediment stress. Similarly, further monitoring and studies of coral reefs succession at Cobourg after catastrophic coral bleaching in 2002–2003 are needed to assess the effect of common high turbidity and sedimentation on corals recovering after mass mortality.

The chosen monitoring design proved to be an accurate tool for measuring changes in live coral cover after a short, minor bleaching event in 2001. However, rapid dramatic changes in coral reef environment caused by catastrophic coral bleaching highlight the necessity to implement transect methods like linear point intercept (LPI) (Beenaerts and Berghe 2005; Nadon and Stirling 2006) in addition to the permanent quadrat method to assess the damage to the reef on the medium scale.

Comparison of data and observation of bleaching obtained *in situ* with the satellite data on surface water temperature and coral thermal stress (Fig. 4) from Coral Reef Watch (CRW) of the National Oceanic and Atmospheric Administration, affirms that this remote sensing system is an important and useful tool to predict and warn of coral bleaching. CRW accurately detected the major bleaching in 2002–2003. Further development of CRW operational products and implementing higher resolution (mapping at 9 instead of current 50 km resolution (Strong *et al.* 2004)) would undoubtedly increase the accuracy of early warning predictions.

The threat of adverse changes to inshore marine ecosystems due to global warming makes coral monitoring absolutely crucial as an 'early warning system'. Establishing new sites for coral monitoring with simultaneous water temperature measurements using *in situ* data loggers in Darwin Harbour, at Cobourg, Gove/Nhulunbuy and at Groote Eylandt is absolutely essential.

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